Salt Marsh Restoration in Connecticut: 20 Years of Science and Management

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Abstract

In 1980 the State of Connecticut began a tidal marsh restoration program targeting systems degraded by tidal restrictions and impoundments. Such marshes become dominated by common reed grass (*Phragmites australis*) and cattail (*Typha angustifolia* and *T. latifolia*), with little ecological connection to Long Island Sound. The management and scientific hypothesis was that returning tidal action, reconnecting marshes to Long Island Sound, would set these systems on a recovery trajectory. Specific restoration targets (i.e., predisturbance conditions or particular reference marshes) were considered unrealistic. However, it was expected that with time restored tides would return ecological functions and attributes characteristic of fully functioning tidal salt marshes. Here we report results of this program at nine separate sites within six marsh systems along 110 km of Long Island Sound shoreline, with restoration times of 5 to 21 years. Biotic parameters assessed include vegetation, macroinvertebrates, and use by fish and birds. Abiotic factors studied were soil salinity, elevation and tidal flooding, and soil water table depth. Sites fell into two categories of vegetation recovery: slow, ca. 0.5%, or fast, more than 5% of total area per year. Although total cover and frequency of salt marsh angiosperms was positively related to soil salinity, and reed grass stand parameters negatively so, fast versus slow recovery rates could not be attributed to salinity. Instead, rates appear to reflect differences in tidal flooding. Rapid recovery was characterized by lower elevations, greater hydroperiods, and higher soil water tables. Recovery of other biotic attributes and functions does not necessarily parallel those for vegetation. At the longest studied system (rapid vegetation recovery) the high marsh snail *Melampus bidentatus* took two decades to reach densities comparable with a nearby reference marsh, whereas the amphipod *Orchestia grillus* was well established on a slow-recovery marsh, reed grass dominated after 9 years. Typical fish species assemblages were found in restoration site creeks and ditches within 5 years. Gut contents of fish in ditches and on the high marsh suggest that use of restored marsh as foraging areas may require up to 15 years to reach equivalence with reference sites. Bird species that specialize in salt marshes require appropriate vegetation; on the oldest restoration site, breeding populations comparable with reference marshland had become established after 15 years. Use of restoration sites by birds considered marsh generalists was initially high and was still nearly twice that of reference areas even after 20 years. Herons, egrets, and migratory shorebirds used restoration areas extensively. These results support our prediction that returning tides will set degraded marshes on trajectories that can bring essentially full restoration of ecological functions. This can occur within two decades, although reduced tidal action can delay restoration of some functions. With this success, Connecticut's Department of Environmental Protection established a dedicated Wetland Restoration Unit. As of 1999 tides have been restored at 57 separate sites along the Connecticut coast.

Key words: ***Fundulus*, *Melampus*, *Phragmites*, restoration, salt marsh, *Spartina*, tidal marsh.**

Introduction

The deep waters of Long Island and Fishers Island Sounds, combined with the glacial history and bedrock topography of Connecticut's shoreline, has limited development of extensive tidal wetlands. In 1880 the state had ca. 8,443 ha of tidal marsh along its 170-km

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coast (Rozsa 1995a); approximately 30% of this area had been lost to fill and dredging before passage of Connecticut's initial Tidal Wetland Act (TWA) in 1969 (Connecticut General Statutes 22a-28-35). This act has effectively preserved the state's remaining tidelands; present permitted loss averages less than 0.1 ha/yr. The total area of tidal wetland in Connecticut is now ca. 5,900 ha, approximately two-thirds of which is *Spartina* (cord grass)-dominated polyhaline salt marsh.

Despite the TWA's effectiveness in protecting tidal marshes, ecological functions in many systems had become degraded as the result of historic alterations in tidal hydrology. Such changes resulted most commonly from tide gates, undersized culverts associated with road and rail causeways, and impoundments for wildlife management and tidal mills.

Diked/drained wetlands, associated with tide-gated mosquito or flood control projects, suffer multiple ecological impacts. First, ecological functions, which depend on tidal linkage between marshland and Long Island Sound, including nutrient processing, sediment trapping, and nursery habitat (Kneib 1997), are decreased or completely lost. Second, the conversion of *Spartina* salt marsh to brackish or fresh marsh, dominated by near monocultures of reed grass (*Phragmites australis*, hereafter referred to as *Phragmites*) (Roman et al. 1984) alters habitat structure, reducing or eliminating use by several tidal marsh-dependent bird species and muskrats (Benoit & Askins 1999). Third, a sharp reduction or elimination of salt marsh invertebrates that depend on tides is seen. Finally, acid sulphate soils are created, converting marshes into non-point sources of pollution (Dent 1986; Portnoy 1991, 1995; Portnoy & Giblin 1997). Undersized culverts reduce tidal prism and can drive similar changes; they may also impound freshwater run-off from surrounding uplands.

Diked/flooded marshes include over 210 ha of wildlife impoundments and mill ponds. Dikes and water control structures eliminate or significantly reduce tides. Retained water from surrounding uplands initially creates shallow open water impoundments, which ultimately become emergent freshwater systems dominated by *Typha* (cattail) spp. and *Phragmites* (Hebbard 1976). In the case of tidal mill ponds tidal flooding occurs, but the tide gates close on the ebb, significantly raising the level of low tide, causing retreat of emergent wetland to "high ground" and creating a characteristic narrow "bathtub ring" of fringing wetland vegetation. Tide mills have not operated for more than half a century, but water control structures still influence several historic mill ponds, particularly in the high-tide-range western Long Island Sound.

The Connecticut Department of Environmental Protection (DEP) Coastal Area Management Program (now the Office of Long Island Sound Programs) determined that the TWA had no provision to deal with marsh degradation caused by such historic hydrologic modifications. This was addressed by the Connecticut Coastal Management Act (CMA) of 1980, which established a policy "to encourage the restoration and rehabilitation of degraded tidal wetland" (Connecticut General Statutes 22a-90-110). This act became the foundation for DEP's 20-year-old tidal marsh restoration program. The earliest project was on the Barn Island marshes (Miller & Egler 1950; Warren & Niering 1993) at the easternmost end of Long Island Sound (mean tide range, 0.8 m). Between 1946 and 1966 five drowned valley marshes within this tideland complex were diked, impounding ca. 52 ha of salt and brackish marsh. The first, impoundment 1 (IP1), converted 20 ha of salt marsh into a freshwater cattail (*Typha* spp.) marsh. In 1978 the dike was breached with a 1.5-m culvert and the tide gate removed from an existing 0.6-m culvert. In 1982 an additional 2.1-m diameter culvert restored essentially full tidal action above the dike. IP1 has become a keystone restoration site in Connecticut for several reasons. Pre-diking vegetation studies and mapping were conducted in the late 1940s (Miller & Egler 1950), and vegetation within the impoundment was described just before installation of the 1978 culvert (Hebbard 1976). Research on the IP1 site, funded in large part by DEP, provides the longest and most complete data set available on the results of recovering tides to diked tidal wetlands.

In this work we test a two-part hypothesis, first at IP1 and subsequently at many additional sites: (1) the structure and functioning of tidal salt marshes are ultimately organized by the tides and (2) returning tidal action to a diked degraded marsh will reconnect the wetland to the estuary and reset the system on a trajectory that will, over time, result in a self-maintaining tidal salt marsh. The final form and function of such tidally restored wetlands cannot be forecast in detail but will reflect biological, chemical, and physical changes associated with historical degradation of ecosystem functions and structure, interacting with the restored tidal hydrology.

The first 15 years of research on angiosperm (Sinicrope et al. 1990; Barrett & Niering 1993), fish (Allen et al. 1994), macroinvertebrates (Fell et al. 1991; Peck et al. 1994), and birds (Brawley et al. 1998) at Barn Island, summarized by Fell et al. (2000), strongly support this central scientific and management hypothesis—the return of appropriate tidal action will restore the ecological functions characteristic of tidal salt marsh communities to marshland degraded by tidal restriction. Also supporting this hypothesis are the findings of Burdick et al. (1997) in *Phragmites*-dominated diked/drained systems from Maine and New Hampshire and other less fully documented Long Island Sound sites (Rozsa 1995b). These results are the basis for an aggressive program of salt marsh restoration by the DEP.

We report here on research that has focused on (1) rates and patterns at which various ecological functions

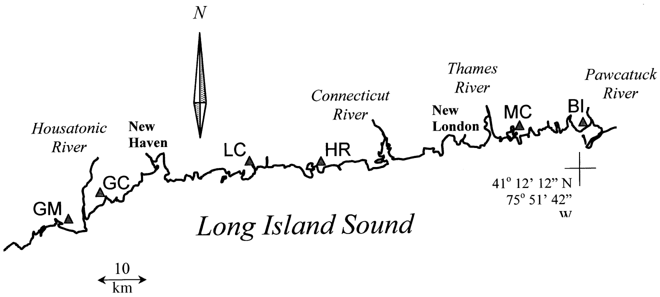

Figure 1. Connecticut coast of Long Island and Fisher's Island Sounds with location of restoration sites included in this study. From west to east, GM, Great Meadows, Stratford; GC, Great Creek, Milford; LC, Long Cove, Gilford; HR, Hammock River, Clinton; MC, Mumford Cove, Groton; BI, Barn Island, Stonington. See Table 1 for site details.

and attributes—specifically support of angiosperm and macroinvertebrate communities, fish use of marsh and creeks, and use of marshlands by birds—return with tidal restoration and (2) how these rates and patterns may be influenced by the two key abiotic factors driving the biology: hydroperiod and salinity. We also report on how these findings have been integrated into restoration management practices by the Connecticut DEP.

Methods

Study Sites

Nine sites within six marsh systems, distributed along 110 km of Long Island Sound shoreline, are included in this study (Fig. 1). Tide ranges, area, nature and date of tidal restriction, restoration date, and restoration approach for each site are summarized in Table 1.

Vegetation

Vegetation recovery after tidal restoration was determined at six of the nine sites, from east to west: Barn Island impoundment 4 (IP4), Mumford Cove (MC), Hammock River (HR), Long Cove (LC), Great Creek (GC), and Great Meadows (GM) (Fig. 1, Table 1). In 1996 vegetation was sampled at five of these locations (all except MC), together referred to as "1996 sites." We established three 40- to 60-m transects at each site, set approximately 90 degrees to tidal creeks and, wherever possible, extending to upland. At Barn Island IP4 and HR, we could also locate an additional control transect in a contiguous reference marsh immediately below the restriction. MC transects were first established in 1992 (Waters 1995) and resampled in 1997. Vegetation was sampled in contiguous 1-m^2 quadrats along each transect, recording species present and visually estimating percent cover within each quadrat.

Vegetation Recovery Rates

The extent of *Phragmites* versus *Spartina*-dominated salt marsh at the 1996 sites was determined through interpretation and planimetery of Connecticut DEP false color infrared aerial photos from 1980, 1986, 1990, and 1995. In addition, in September 1996 *Phragmites* stem densities, separated as "live" (current year's growth) and "dead" (standing prior year's growth), were counted in three 0.25-m^2 quadrats located within a 2-m radius of

Table 1. Salt marsh tidal restoration sites included in this study

Restoration Site	*Mean Tide Range (m)*	*Cause of Degradation or Tidal Restriction*	*Year Restricted*	*Restoration Approach*	*Year Opened*	*Restored Area (ha)*
Barn Island 4	0.87	Diked impounded	1947	Dike breach w/1.3-m culvert	1987	3.4
Barn Island 3	0.87	Diked impounded	1947	Dike breach w/ 1.5-m culvert; 0.6-m culvert opened	1991	11.1
Barn Island 2	0.87	Diked impounded	1946	Dike breach w/ 1.5-m culvert; 0.6-m culvert opened	1978	5.2
Barn Island 1	0.87	Diked impounded	1947	Dike breach w/ 1.5-m (1978) and 2.1-m (1982) culverts; 0.6-m culvert opened	1978-82	27.6
Hammock River	1.54	Diked and drained—tide gates	ca. 1900	Open 1 of 4 tide gates (1.2 × 1.2 m)	1985	46.5
Long Cove	1.71	Diked and drained—undersized culvert clogged ditches and creeks	1939	Open abandonded 1.3-m culvert; clean 2-m ditch	1987	15.7
Great Creek	2.07	Diked and drained—undersized culvert	pre 1936	Construct new 5-m creek to Sound; install two 1.8-m self-regulating tide gates	1986	29.6
Great Meadows	2.13	Diked and drained—tide gates	ca. 1940	Three 1.8-m tide gates abandonded	1990	31.2
Mumford Cove	0.77	Diked and filled dredge spoil site	1954	Fill removed; dike breach w/2-m restored creek and new 1-m branches	1990	15

each soil water well on every transect at IP4, HR, LC, and GC. Mean height was calculated for 12 randomly selected shoots, 4 from each quadrat. If there were less than four shoots in a quadrat, the difference was made up from the other two. All shoots were averaged if there were less than 12. Recovery rate at MC was estimated from 1992 and 1997 transect data and at IP1 using data from Sinicrope et al. (1990) and Barrett and Niering (1993).

Salinity

Soil water wells (open-bottom, 3.8-cm, plastic pipe perforated at 5-cm intervals starting 5 cm below the marsh surface, set 0.3 m into the peat) were established at 3, 10, and 30 or 40 m along each 1996 transect. Well water salinity, measured with a refractometer ($\pm 0.5\%$), and water table depth (± 0.5 cm) were measured on alternate weeks from mid-June to mid-August. Surface peat salinity was also measured in water squeezed from a 1.5×5-cm core taken within a 1-m radius of the wells.

Elevation

Elevations were determined with an optical level at 1-m intervals along all transect lines. Site and transect locations of the 1996 sites did not allow elevations to be set to a common established benchmark. Rather, elevations were set relative to estimated mean high water at each transect, taken as the creek bank elevation where low marsh *Spartina alterniflora* cover fell to less than 10%. Within individual transects, therefore, relative elevations of *Phragmites*-dominated ($>20\%$ *Phragmites* cover) points can be compared with those with low *Phragmites* cover ($<20\%$), but absolute elevation comparisons cannot be made among transects. MC elevations were set relative to local mean lower low water, measured on site in 1992 (Waters 1995).

Macroinvertebrates

Macroinvertebrates were sampled at the five 1996 sites with triplicate 0.25-m^2 quadrats located at 3, 10, and 30 or 40 m along each vegetation transect. Details of the sampling procedure have been described previously (Fell et al. 1998). Animals were collected using a 0.25-m^2 (50×50-cm) wooden sampling frame, 9 cm high, anchored to the marsh surface at each corner. Vegetation within the frame was examined for the presence of animals and then clipped at the bases of stems to facilitate collecting of animals within the litter and on peat surface. An attempt was made to collect all macroinvertebrates observed within the quadrats, but some more active animals were able to escape. Collecting from the periphery of the quadrat toward the center, with two people working each quadrat, minimized such loss. The MC marsh was sampled in 1998, using 0.25-m^2 quadrats situated 5 m apart along transects established in areas dominated either by *Spartina alterniflora* (38 quadrats) or stunted *Phragmites australis* (25 quadrats).

With the same sampling technique macroinvertebrates were sampled in the summer of 1996 above and below the dikes at Barn Island IP2, IP3, and IP4 (tidal flooding restored for 18, 5, and 9 years, respectively) using 0.25-m^2 quadrats situated 5 m apart along transects set normal to tidal creeks. Marsh above and below the IP1 dike was similarly sampled in 1999 along transects previously sampled in 1990 (Fell et al. 1991).

Fish and Macrocrustaceans

In 1995 fish were caught in unbaited Breder traps on the flooded marsh surface at Barn Island IP1 and, as a reference site, the Headquarters (HQ) marsh situated immediately below the impoundment dike. The trap was a plexiglas box, $31 \times 16 \times 15$ cm, with a vertical slit-like opening, 1.3 cm wide, extending from the top to the bottom at the back of a funnel that is 28 cm wide at its mouth (Breder 1960). The traps were placed 5 m apart in a line parallel to and about 4 m back from a ditch bank (Fell et al. 1998). From early February through mid-November 1999 mosquito control ditches and the tidal creek in these two areas were sampled with unbaited Bell minnow traps (Bell Distributors Ltd., South Haven, MI). Six to 12 traps were set in each marsh and left over two tidal cycles, usually at 1-week intervals for a total of 33 sampling days.

At MC fish and crustaceans were trapped in reestablished creeks using a Fyke net (Wilcox Marine, Mystic, CT) with wings extending to each bank (Allen et al. 1994). Animals were collected during four ebbing tides in June and July 1998.

Diet composition of the fish *Fundulus heteroclitus* (mummichog) trapped on the Barn Island marshes was determined as described by Allen et al. (1994). The relative volume of every food type in each gut (sections I and II, Babkin & Bowie 1928) was estimated visually and scored as either more than or less than 50% of the total gut content volume. A gut fullness index (the wet weight of the pooled gut contents expressed as a percentage of the total wet body weight of the fish) was calculated for each sample of fish caught while leaving the marsh with the ebbing tide.

Birds

During the summers of 1994 and 1995 bird surveys were conducted at MC (Brawley 1995), Barn Island IP1 and IP3, and HQ, the reference site below the IP1 dike

(Brawley et al. 1998). In each marsh, location and behavior of all birds seen or heard were recorded within a 25 × 100-m plot. Thirty-minute surveys were conducted a total of eight times (four times a year) in each marsh between May and August. Birds feeding in the air above the plot were also recorded, but not transient individuals passing over the plot. Species recorded were divided into four groups: marsh specialists (species dependent on tidal marshes for breeding, including Willet, Marsh Wren, Saltmarsh Sharp-tailed Sparrow, and Seaside Sparrow), long-legged waders (Herons and Egrets), shorebirds (Sandpipers), and marsh generalists (ubiquitous species that may forage and breed in uplands and tidal wetlands, including Common Yellowthroat, Song Sparrow, and Red-winged Blackbird).

In 1999 plots were resurveyed using 10-minute fixed radius point counts. All species were identified either visually or aurally within 50 m, between 50 and 100 m, and greater than 100 m from a plot's center and reported by group type. For comparison of data with the earlier study only species recorded within 100 m of the point were included in this analysis. The point counts were conducted three times at each site during the months of May and June. Observations were summarized as the mean number of individuals recorded per visit for all sites during survey periods.

To assess changes in bird occurrence at the restoration marshes over time, relative abundance (numbers at the restoration sites divided by those at HQ) of marsh specialists and marsh generalists at both sampling periods were plotted against years of restoration at the three restoration sites. In 1995 these sites had been under restoration for 4 (IP3), 5 (MC), and 15 (IP1) years and in 1999 for 8 (IP3), 9 (MC), and 19 (IP1) years.

Results

Vegetation Recovery

From the false color infrared aerial photos there is an order-of-magnitude difference in the rates at which salt marsh vegetation replaced *Phragmites* at the HR (11 years of restoration), LC (10 years), and GM (5 years) sites (5 to 7%/yr) versus IP4 (9 years) and GC (6 years) (ca. 0.5%/yr) (Table 2). In addition, rates of vegetation recovery had no apparent relationship with the age of tidal restoration.

Vegetation recovery was also about 5%/yr at both IP1 and MC. Using 1988 data (Sinicrope et al. 1990; Barrett & Niering 1993) salt marsh vegetation at IP1 replaced cattail at ca. 5.5%/yr over the first decade of tidal restoration. During that time *Phragmites* cover actually increased as it colonized areas opened by the loss of cattail. By 1999, 21 years after the initial reestablishment of tidal flow, IP1 was dominated by stunted *Spartina alterniflora* (smooth cord grass) with significant areas of *S. patens* (salt meadow cord grass) and *Distichlis spicata* (spike grass), particularly along creek bank levees. *Phragmites* occurred principally as a band along the upland, where it was frequently stunted and still losing dominance to *S. alterniflora* (unpublished data). Recovery slowed over the second decade, however; based on field observations and 2000 aerial photographs, cattail and *Phragmites* covered approximately 20% of IP1.

Table 2. Recovery of salt marsh vegetation after tidal restoration measured as the loss of *Phragmites* cover up to 1995, the latest complete false color infrared air photo set available.

Marsh System	*Year Opened*	*Restoration Area (ha)*	*% Phragmites Dominated*	*Recovery Rate (%/yr)*
Hammock River	1984	46.5	43	5.2
Great Creek	1986	29.6	97	0.3
Long Cove	1987	15.7	31	8.6
Barn Island	1987	3.4	96	0.5
Great Meadows	1990	31.2	69	6.2

All sites were 100% *Phragmites* before tidal restoration.

In the spring of 1990 the fill had just been removed from the MC restoration site and the substrate surface had virtually no angiosperm cover. In 1992 mean *S. alterniflora* cover along transects was 11%; by 1997 *S. alterniflora coverage* was 52% (Fig. 2a), an average increase of 7.6 %/yr.

Salinity. Mean salinities of well water versus nearby surface peat did not differ for any of the 51 wells at the 1996 sites (IP4, HR, LC, GC, and GM; paired *t*-test, well vs. peat for all observations: $p < 0.001$). Salinities and water table depths among transects varied significantly within each site except GM (analysis of variance, $p < 0.05$), but pooling all wells within a marsh there were no well or peat salinity differences among the five restoration sites (15 transects, three wells each: $p = 0.75$). There also were no significant salinity differences by any measure (individual readings, well means, transect means, site means) between the three rapidly recovering and two slowly recovering sites (Table 3).

Soil salinity was correlated with salt marsh vegetation and *Phragmites* cover. Cover or frequencies of individual salt marsh species (*S. alterniflora*, *Spartina patens*, *D. spicata*, *Juncus gerardii* [black grass], and pooled forbs) were not significantly related to salinity. When cover was combined and frequencies averaged, however, correlations were significant (Fig. 3a). Similarly, *Phragmites* cover and frequency (Fig. 3b) were negatively correlated with salinity. *Phragmites* disappears abruptly above 26% but shows some variability at

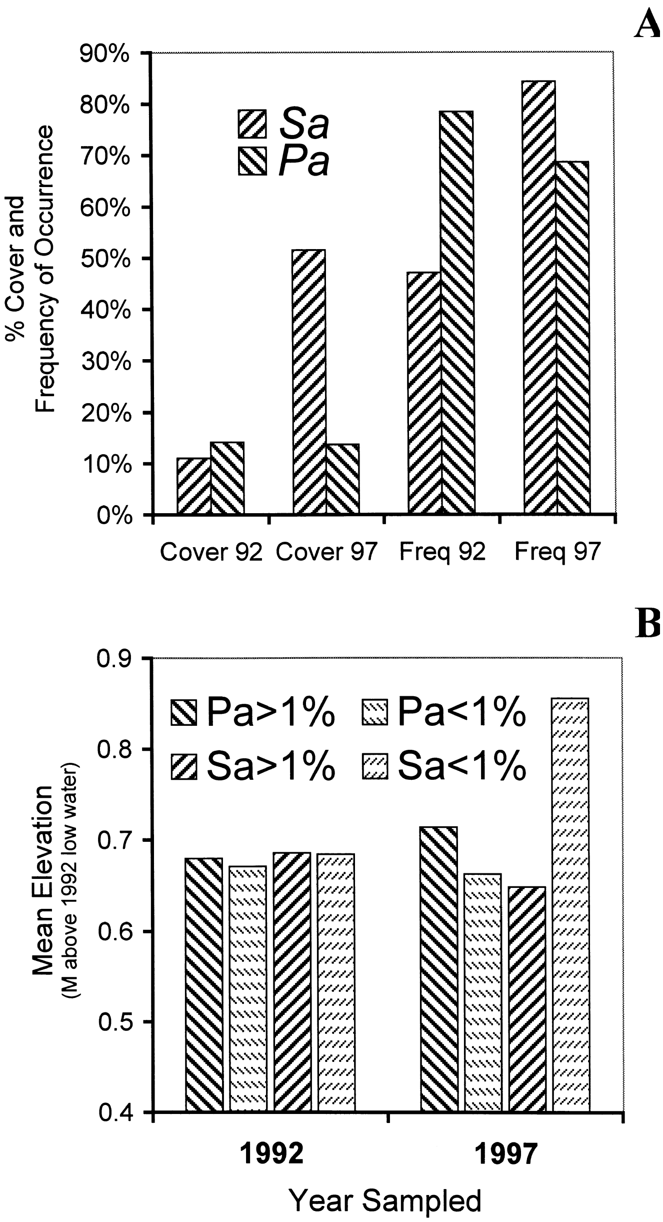

Figure 2. (A) Mean percent cover and frequency of occurrence of *Spartina alterniflora* (*Sa*) and *Phragmites australis* (*Pa*) along three Mumford Cove transects sampled in 1992 and 1997, 2 and 7 years after restoration. Cover (Tukey's test $p > 0.05$) and frequency (chi-square $p > 0.05$) increased for *Sa* but not for *Pa.* (B) Mean elevations (datum = 1992 local mean lower low water) of points along three Mumford Cove transects sampled in 1992 and 1997 that supported (cover ≥ 1%) *Sa* and *Pa* and points that were essentially free of these species (cover < 1%).

slightly lower salinities. This same pattern occurs with *Phragmites* stem height ($r^2 = 0.46$, $p > 0.001$) (Fig. 3c), and percent stems flowering ($r^2 = 0.25$, $p = 0.006$; data not shown) but not stem density. Height, density, and flowering all show a good deal of scatter below 26%, suggesting factors in addition to salinity are influencing loss of *Phragmites.*

Water Table. Water table depths among the five restoration sites were significantly different (analysis of variance, $p = 0.04$). Depth to water table was less by any measure (individual readings, well means, transect means, site means) when comparing the three rapidly recovering sites with the two slower ones (*t*-test unequal variances, $p = 0.04$ for transect means, $p \leq 0.001$ by other measures) (Table 3). *Phragmites* stem density increased as the water table dropped ($r^2 = 0.22$, $p = 0.01$; data not shown), but no other plant parameters had any statistical relationship with this factor.

Elevations. Of the 15 transects behind breached dikes there was one on which *Phragmites* cover never exceeded 20% and a second on which *Phragmites* cover always exceeded 20%. On the remaining 13 mean elevations of *Phragmites*-dominated points were significantly higher on eight transects (*t*-tests unequal variances, $p < 0.05$); they were lower on two lines with no difference on the remaining three. Using transect means as individual observations, elevations of points still dominated by *Phragmites* in the rapidly recovering three systems were significantly higher than those with low or no *Phragmites* cover ($\Delta = 3.9$ cm, $p = 0.03$ by paired *t*-test). On the two more slowly recovering systems, however, the difference between points dominated by *Phragmites* and points of low *Phragmites* cover was not significant ($\Delta = 1.7$ cm, $p = 0.49$ by paired *t*-test).

In the spring of 1990 MC was essentially bare peat. By 1992 both *S. alterniflora* and *Phragmites* were sparse but uniformly established, with distribution apparently unrelated to elevation. By 1997, however, mean elevation of *S. alterniflora* points was significantly lower than those without this species and those with *Phragmites* cover more than 1% (Fig. 2b).

Macroinvertebrate Recovery

Barn Island. In 1999, 21 years after reestablishment of tidal flow, the pulmonate gastropod *Melampus bidentatus* (coffee bean snail) was as abundant in IP1 as in the reference marshes below the dike (HQ) (Fig. 4), but mean densities of the gammarid amphipod *Orchestia grillus* (hereafter *Orchestia*) and the isopod *Philoscia vittata* (hereafter *Philosica*), which prefer higher marsh elevations (Fell et al. 1982; Kneib 1982), and of the gammarid amphipod *Uhlorchestia spartinophila* (hereafter *Uhlorchestia*) were lower (Table 4). Furthermore, densities of the low marsh amphipod *Gammarus palustris* (hereafter *Gammarus*) were higher in the recovering marsh above the dike, dominated by stunted *S. alterniflora*, than on the reference marshes.

Table 3. Mean recovery rates, seasonal soil water well and peat salinities, and depths to water table for the three rapidly recovering (Hammock River, Long Cove, and Great Meadows) and two slowly recovering (Barn Island and Great Creek) systems.

Marsh Systems	*Mean Recovery Rate (% Phragmites loss/yr)*	*Well Salinity (%)*	*Peat Salinity (%)*	*Depth to Water Table* (cm)*
Hammock River				
Long Cove	6.1	22.9 ± 2.3	23.6 ± 3.6	24.2 ± 1.9
Great Meadows				
Barn Island				
Great Creek	0.5	22.7 ± 1.0	21.7 ± 1.1	28.8 ± 0.9

Salinity and water table values are means from transect season means (three wells/transect sampled three to five times from June to August, three transects/system). There are no significant salinity differences between rapidly recovering and slower systems measured from wells or peat (*t*-test: unequal variances, $p > 0.75$). Depth to water table was significantly greater (*t*-test: unequal variances, $p = 0.04$) on the two slower systems.
* Means are significantly different; $p = 0.04$.

In contrast, in 1996, after 9 years of restoration, IP4 was still largely dominated by stunted *Phragmites*, but mean densities of *Philoscia*, *Orchestia*, and *Gammarus* were comparable with those of the reference marsh below the dike. *Melampus* and *Uhlorchestia* densities, however, were still significantly lower above the dike than below.

In 1996, 5 years after tidal restoration to Barn Island IP3, roughly half the area was dominated by *S. alterniflora*, but there were still large sparsely vegetated patches with shallow (1–5 cm) standing water. This recovering marsh possessed a typical assemblage of macroinvertebrates, but most at significantly lower densities than in the reference marsh below the dike (Table 4). For exam-

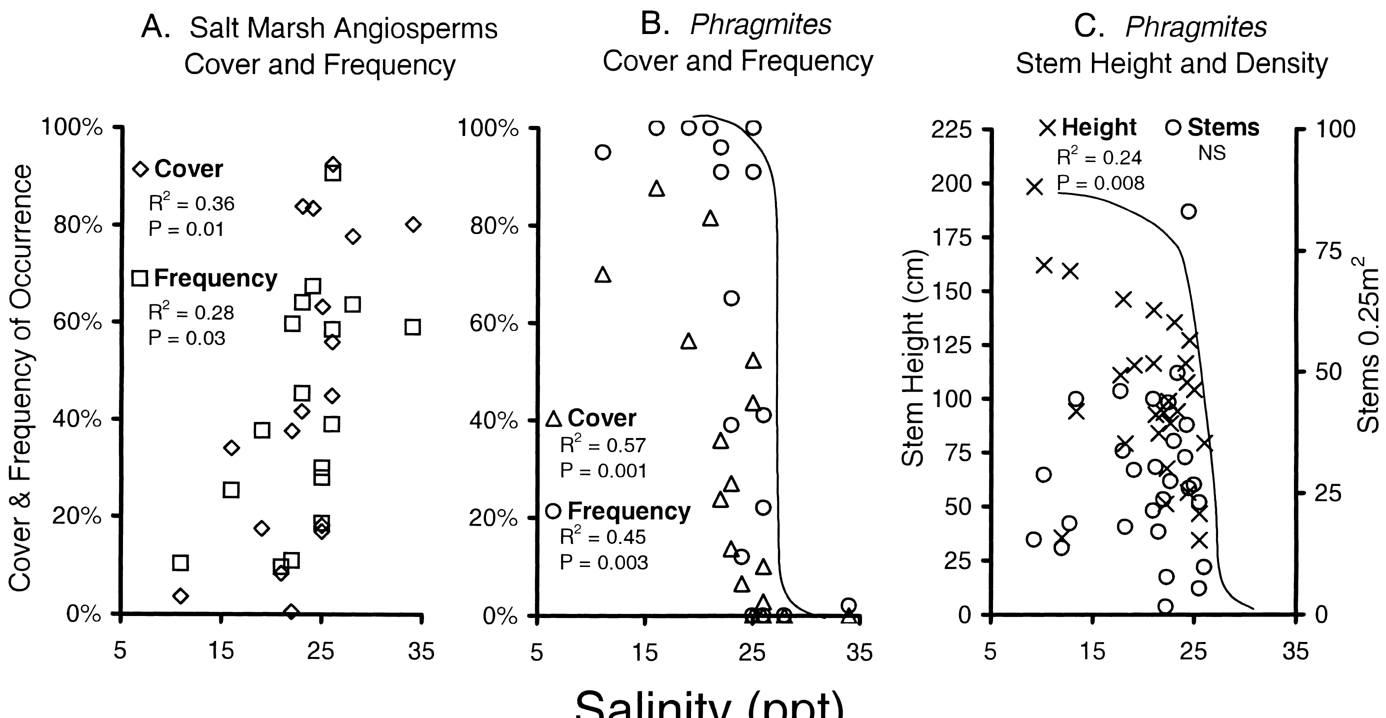

Figure 3. (A) Total mean percent cover and frequency of occurrence for all salt marsh angiosperms along 1996 restoration transects versus transect mean salinity. Both measures increase with salinity and regressions are significant. (B) Mean percent cover and frequency of occurrence for *Phragmites australis* along 1996 restoration transects versus transect mean salinity. Both measures decrease with salinity and regressions are significant. Frequency drops sharply above 26%; curve fitted by hand. (C) Mean end of season height and stem density of *Phragmites australis* at transect soil water wells ($n = 27$). Height drops with salinity with maximum salinity ca. 26%; curve fitted by hand. Stem density does not correlate with salinity.

Table 4. Mean density (no./m^2 ± SE) of six macroinvertebrates in recovering and reference regions of three marshes at Barn Island, Connecticut that have been in the process of restoration for different periods of time.

	IP3 (5 yr, 1996)		*IP4 (9 yr, 1996)*		*IP1 (21 yr, 1999)*	
	Recovering	*Reference*	*Recovering*	*Reference*	*Recovering*	*Reference*
Species	$n = 26$	$n = 18$	$n = 19$	$n = 19$	$n = 76$	$n = 60$
Melampus (snail)	128 ± 32 (69)	1280 ± 168 (100)	176 ± 56 (74)	616 ± 112 (95)	**584 ± 56 (88)**	**489 ± 56 (90)**
Geukensia (mussel)	6.8 ± 2.8 (38)	25.2 ± 9.2 (72)	0.8 ± 0.8 (5)	3.2 ± 1.2 (26)	**2.8 ± 1.3 (17)**	**4.2 ± 1.1 (37)**
Philoscia (isopod)	2.0 ± 1.6 (12)	7.6 ± 2.4 (50)	**44 ± 12.8 (79)**	**47 ± 16 (58)**	4.0 ± 1.3 (25)	17.4 ± 5.9 (37)
Orchestia (amphipod)	2.8 ± 2.4 (8)	16.4 ± 4.4 (78)	**48 ± 8.0 (100)**	**50 ± 8.4 (95)**	10.4 ± 1.9 (45)	37.8 ± 7.7 (80)
Uhlorchestia (amphipod)	76 ± 16.4 (81)	168 ± 19.2 (100)	40 ± 9.2 (89)	132 ± 27.2 (100)	67.8 ± 8.6 (76)	112.9 ± 17.6 (92)
Gammarus (amphipod)	264 ± 68.0 (65)	2.8 ± 2.0 (11)	**0.8 ± 0.4 (11)**	**0.4 ± 0.4 (5)**	44.3 ± 8.7 (45)	5.5 ± 2.3 (15)

Frequency (%) of occurrence is in parentheses. n = the number of quadrats sampled. Bold pairs are not significantly different (t-tests, $p \leq 0.05$).

ple, even in areas of IP3 dominated by *S. alterniflora*, density of *Melampus bidentatus* was only 15% that of comparable vegetation below the dike. An exception was the amphipod *Gammarus*, which was more abundant in IP3, especially in relatively wet areas with dense plant cover.

Although *Melampus* occurred at a low mean density in IP3, it tended to reach larger sizes compared with the reference marsh below the dike. In IP3 69% of the snails were more than 8 mm in shell length, whereas in the reference marsh only 27% exceeded this size. However, there was a prominent group of small snails (2.1–5.0 mm) in IP3 that had presumably settled on the marsh during the previous summer.

1996 Sites. As with total salt marsh angiosperm cover, mean density of *Melampus* along transects was significantly correlated with surface peat salinity ($r^2 = 0.28$, $p = 0.04$). None of the other species demonstrated any significant relationship with salinity, and they were quite independent of each other (Table 5). Except for the isopod *Philosica* at HR, densities of all species on the two reference transects (IP4 and HR) were greater than those in the comparable restoration areas. It is also important to note that although marshes were judged to be recovering rapidly or more slowly based on vegetation, these designations do not necessarily apply to reestablishment of some macroinvertebrate populations (Table 6).

Mumford Cove. By 1998, 8 years after tidal restoration, macroinvertebrate populations had become well established (Table 7), but mean densities for most species tended to be lower than typically found on reference systems (see reference marshes in Table 4 and

Table 5. Macroinvertebrate densities along 1996 site transects.

			Animals/m^2			
Marsh System	*Transect*	*Peat Salinity (%)**	Melampus bidentatus	Orchestia/Uhlorchestia *(amphipods)*	Philoscia vittata *(isopod)*	Hydrobia *spp.*
Barn Island IP4	C	33a	332	176	69	0
	1	23b	269	91	44	0
	2	18c	122	66	63	0
	3	19bc	8	113	19	0
Great Creek	1	24a	439	24	0	0
	2	23a	104	40	0	3
	3	23a	437	48	0	1
	C	29a	2572	23	36	0
Hammock River	1	23a	353	15	63	47
	2	22b	451	1	23	359
	3	23a	240	1	25	19
Long Cove	1	10a	269	72	0	0
	2	44b	837	16	0	1
	3	10a	129	70	14	138
Great Meadows	1	26a	85	9	0	245
	2	27a	63	0	0	28
	3	27a	12	1	0	11

Densities are means from three samples per transect taken near soil water wells. C = control transects, established below the tidal restriction.
*Within systems, transects followed by the same letter are not significantly different (Tukey's test, $p < 0.05$).

Table 6. Macroinvertebrate densities (mean animals/m^2 ± SE) between the two slower and three faster recovering 1996 restoration sites compared by *t*-test (equal variances for all but *Hydrobea*).

		Animals/m^2			
	Restoration Sites	Melampus bidentatus	Orchestia/Uhlorchestia	Philosica vittata	Hydrobea *spp.*
Slower $n = 6$	Barn Island Great Creek	229.8 ± 81.2	63.7 ± 14.9	21.0 ± 12.0	0.7 ± 0.5
Faster $n = 9$	Hammock River Long Cove Great Meadows	271.0 + 90.6	20.6 + 10.3	13.9 ± 7.5	94.2 ± 45.3
t-test $p =$		0.740	0.020	0.576	0.060

control transects in Table 5). Also, there were some sharp differences in mean densities between areas dominated by *S. alterniflora* versus stunted *Phragmites. Melampus, Uhlorchestia*, and *Gammarus* were much more abundant in *Spartina*-dominated areas, whereas *Orchestia* and the isopods *Philosica, Trachelipus rathkei*, and *Porcellio* sp. occurred at greater densities within stunted *Phragmites* (Table 6).

Fish

Barn Island. In 1999 *F. heteroclitus* represented 95% of the 53,295 fish caught and was as abundant in IP1 as in HQ, the reference marsh below the dike (Table 8). In fact, the mean number of *F. heteroclitus* caught per trap per day in mosquito-control ditches was greater in IP1 (127 ± 8) than in HQ (101 ± 6) ($t = 2.818$, $df = 32$, $p = 0.008$), whereas the mean numbers caught per trap per day in the tide creek were not different above (62 ± 3) and below (61 ± 5) the dike ($t = 0.228$, $df = 32$, $p = 0.821$). Furthermore, the mean species richness of fish caught in the creek and ditches in IP1 (3.67 ± 1.05) and HQ (3.55 ± 1.12) were not significantly different ($t = 0.56$, $df = 32$, $p = 0.580$). Overall, 10 species of fish and 3 crustaceans were caught in mosquito-control ditches in IP1 compared with 11 and 2, respectively, in HQ. *Cyprinodon variegates* (sheepshead minnow), *Fundulus luciae* (spotfin killifish), and *Anguilla rostrata* (American eel) tended to be more abundant in IP1, whereas *Apeltes quadracus* (fourspine stickleback) tended to be more numerous below the dike.

In 1995 the diets of *F. heteroclitus* caught on flooded marsh surfaces of IP1 and HQ were similar (Table 8). Amphipods, insects, algae, and detritus were prominent components. Interestingly, the marsh-surface *Orchestia*, *Uhlorchestia*, *Philosica*, and *Melampus* were present only in low frequencies in the gut contents of fish caught on both marshes. Gut fullness indices suggest that *F. heteroclitus* in IP1 were ingesting as much food material as those in the reference marsh (Table 9).

Mumford Cove. Eight years after restoration nine different species of fish were caught in Fyke nets blocking reestablished creeks and ditches (Table 7), seven of which were common with those from the 1999 minnow trapping on Barn Island. The Atlantic silverside *Menidia menidia* was the most abundant fish at MC (59% of all fish caught); *F. heteroclitus* was the second most abundant (25% of the total catch).

Birds

During the summers of 1994 and 1995 a diverse assemblage of wetland birds was identified at HQ (reference

Table 7. Mean density of macroinvertebrates (animals/m^2 ± SE) in different regions of the restored Mumford Cove marsh 8 years after reestablishment of tidal flooding.

Species	*Spartina alterniflora Dominated* n = *38*	*Stunted Phragmites Dominated* n = *25*
Melampus bidentatus (snail)	112.0 ± 18.4 (82)	6.0 ± 3.2 (20)
Orchestia grillus (amphipod)	29.6 ± 5.6 (87)	53.2 ± 8.8 (100)
Uhlorchestia spartinophila (amphipod)	100.0 ± 18.8 (95)	0.4 ± 0.4 (8)
Gammarus palustris (amphipod)	31.2 ± 16.8 (32)	0
Philosica vittata (isopod)	0.8 ± 0.4 (11)	34.8 ± 10.0 (60)
Trachelipus rathkei (isopod)	0.4 ± 0.4 (8)	1.6 ± 0.8 (20)
Porcellio sp. (isopod)	0	16.4 ± 7.6 (28)
Oniscus sp. (isopod)	0	1.2 ± 0.8 (3)

Frequency (%) of occurrence is in parentheses; n = number of quadrats sampled.

Table 8. Fishes and crustaceans caught within mosquito control ditches at Barn Island in the recovering marsh (IP1) and the adjacent reference marsh (HQ) below impoundment dike 21 years after restoration and in recreated creeks of the restored Mumford Cove marsh 8 years after return of tidal flooding.

		Total Number Caught		
		Barn Island		
Species	*Common Name*	*IP1*	*HQ*	*Mumford Cove*
Fundulus heteroclitus	Mummichog	16,004	12,677	2,327
Cyprinodon variegatus	Sheepshead minnow	1,134	211	469
Apeltes quadracus	Fourspine stickleback	33	110	
Anguilla rostrata	American eel	68	29	
Fundulus luciae	Spotfin killifish	80	1	
Fundulus majalis	Striped killifish	12	9	926
Pungitius pungitius	Ninespine stickleback	7		5
Lucania parva	Rainwater killifish	4	1	
Menidia menidia	Atlantic silverside	2	1	5,571
Gasterosteus aculeatus	Threespine stickleback	2	2	2
Pholis gunnellus	Rock gunnel		1	
Myoxocephalus scorpius	Shorthorn sculpin		1	
Pleuronectes americanus	Winter flounder			37
Mugai curema	White mullet			34
Syngnathus fuscus	Northern pipefish			1
Carcinus maenas	Green crab	31	56	138
Uca pugnax	Blackback fiddler crab	2		
Palaemonetes pugio	Grass shrimp*	524	638	7161

At Barn Island animals were captured using unbaited minnow traps, three to six per site, each set for 24 hr at ca. weekly intervals from early February to mid-November 1999 (33 total trapping days). At Mumford Cove animals were caught with a 6-mm mesh Fyke net during four ebbing tides (June 10 and 25, July 14 and 24).
*Enumerated beginning in late May at Barn Island.

for the IP1, IP3, and MC) (Table 10), including breeding populations of Willets and Saltmarsh Sharp-tailed Sparrows, both marsh specialists and listed as "Species of Special Concern" by the Connecticut DEP. No long-legged waders were present during the surveys, but shorebirds, particularly Least and Semipalmated Sandpipers, were recorded foraging in shallow pools on the high marsh during fall migration. A similar use pattern

Table 9. Frequency (%) of occurrence of foregut components in *Fundulus heteroclitus* (4.8–9.0 cm total length) trapped on flooded regions of Barn Island IP1 and the reference marsh (HQ), fall of 1995.

	September 28		*October 26*	
Foregut Contents	*Recovering* $n = 40$	*Reference* $n = 39$	*Recovering* $n = 41$	*Reference* $n = 40$
Major components				
Amphipods	28 (8)	33 (13)	**66** (27)	**33** (10)
(*Orchestia/Uhlorchestia*)	10 (5)	8 (8)	10 (7)	5 (5)
Larval and adult insects	60 (18)	36 (10)	34 (5)	43 (20)
Algae	**23** (10)	**64** (31)	29 (10)	30 (8)
Detritus	58 (15)	44 (5)	41 (5)	38 (8)
Selected minor components				
Copepods	13 (0)	10 (0)	32 (0)	15 (0)
Isopods	15 (3)	0	10 (5)	3 (3)
(*Philoscia*)	8 (3)	0	5 (2)	3 (3)
Shrimp	5 (5)	3 (3)	2 (0)	10 (8)
Spiders	5 (0)	3 (3)	7 (5)	18 (8)
Gastropod molluscs (*Melampus*)	8 (5)	0	0	15 (5)
	3 (3)	0	0	8 (5)
Unrecognizable	48 (5)	26 (3)	54 (5)	43 (5)
Gut fullness index	4.05	3.03	1.33	0.87

Frequency with which various components represented more than half of the total gut content volume is given in parentheses. n = the number of fish guts examined. Bold pairs of major gut content components of fish caught at the same time in IP1 and the reference marsh are significantly different at the 0.05 level (2 × 2 chi-square on actual data).

Table 10. Abundance of birds (average number of individuals observed per visit) at the reference marsh, HQ, and the restoration marshes, IP1, IP3 sites at Barn Island and the MC marsh during surveys conducted in the summers of 1994–1995 and 1999.

	1994–1995 (n = *8)*				*1999* (n = *3)*			
Species	*HQ*	*IP1*	*IP3*	*MC*	*HQ*	*IP1*	*IP3*	*MC*
Marsh Specialists								
Willet *(Catoptrophorus semipalmatus)*	1.4			1.3				
Marsh Wren *(Cistothorus palustris)*					1.3	0.3	0.7	
Saltmarsh Sharp-tailed Sparrow *(Ammodramus caudacutus)*	1.1	2.4	0.3		3.7	4.3		1.3
Seaside Sparrow *(Ammodramus maritimus)*		2.8		0.3	1.0			
Long-legged waders								
Great Blue Heron *(Ardea herodias)*								
Great Egret *(Ardea alba)*			0.1	0.3			1.0	
Snowy Egret *(Egretta thula)*		0.1		0.4			0.3	0.7
Green Heron *(Butorides striatus)*				0.1				
Glossy Ibis *(Plegadis falcinellus)*		0.5						
Shorebirds								
Killdeer *(Charadrius vociferus)*		0.1	0.1				1.3	
Greater Yellowlegs *(Tringa melanoleuca)*		0.1				1.0		
Lesser Yellowlegs *(Tringa flavipes)*		1.3						
Spotted Sandpiper *(Actitis macularia)*			0.1	0.3				
Semipalmated Sandpiper *(Calidris pusilla)*	1.1	0.1	0.1					
Least Sandpiper *(Calidris minutilla)*	2.9	3.3	1.8	0.5			0.3	
Baird's Sandpiper *(Calidris bairdii)*		0.3						
Marsh generalists								
Common Yellowthroat *(Geothlypis trichas)*				0.3	1.3	1.7	3.3	0.3
Song Sparrow *(Melospiza melodia)*			0.4	1.1	1.3		2.3	2.0
Red-winged Blackbird *(Agelaius phoeniceus)*	0.5	1.0	2.0	0.8	1.7	5.3	9.7	3.3

n = number of surveys at each site.

occurred during the summer of 1999; abundance (average number recorded per visit) of Saltmarsh Sharp-tailed Sparrows was greater during the latter surveys, however, along with abundance and richness of marsh generalists.

In 1994 and 1995, after approximately 15 years of tidal restoration, IP1 supported a greater abundance and diversity of birds (10 wetland species representing all four groups used here: marsh specialists, waders, shorebirds, and marsh generalists) than any of the other sites investigated. Seaside and Saltmarsh Sharp-tailed Sparrows nested in stunted *S. alterniflora* throughout the marsh. Long-legged waders such as Snowy Egrets and Glossy Ibis and shorebirds such as Greater and Lesser Yellowlegs were also recorded in the shallow pools and pannes on the high marsh (Brawley et al. 1998). The density of marsh specialists present in 1999 was comparable with the earlier surveys. Waders and shorebirds were less abundant during the second survey, likely reflecting the earlier sampling dates in May and June.

During the 1994 and 1995 surveys the IP3 marsh surface was frequently flooded, limiting use of this habitat by ground-nesting marsh specialists. However, these wet conditions and the presence of two large permanent pools near the impoundment dike attracted a suite of shorebirds throughout the summer. The surrounding forest edge and dense *Phragmites* along the marsh periphery provided perching and nesting sites for generalists such as Red-winged Blackbirds and Song Sparrows, which can be abundant in degraded sites. In 1999 the abundance of marsh generalists was greater than in 1994 and 1995, consistent with all the other sites surveyed.

Although different in restoration history, numbers and species of birds at MC in 1994 and 1995 after 5 years of restoration was similar to IP3 after 4 years (Brawley 1995). The presence of several pools attracted a variety of waders and shorebirds, but marsh generalists largely dominated the area. No marsh specialists used MC during the 1994 and 1995 surveys, but by 1999 Saltmarsh Sharp-tailed Sparrows were observed nesting at the site, with an average of 1.3 individuals recorded per visit (Fig. 5).

Connecticut DEP and Tidal Marsh Restoration

Since the mid-1970s tidal restrictions have been removed or modified at 57 separate sites along the Connecticut shore, returning tides to 680 ha of coastal marshland. These sites include tide gates abandoned by the state's Mosquito Control Program and removed by

Table 11. Approaches and methods of tidal restoration at 57 Connecticut DEP sponsored or permitted projects between 1975 and 1999.

Restoration Methods	*Percent of Projects Where Used*
Tide-gate removal	35
Self-regulating gates installed	14
Tide-gate management	5
Culvert resized	28
Outlet/channel dredged/cleaned	19
Fill removal	5

the town of Fairfield in the mid to late 1970s. Since 1978, however, most restoration projects have been performed directly by or with technical assistance from DEP. About two-thirds of these were started after 1980, when the Connecticut CMA provided the statutory basis for DEP's commitment to tidal marsh restoration. By 1990 tides had been returned to many systems that presented few technical and legal challenges. Removal of remaining tidal restrictions became increasingly complex, both technically and legally, and over the last decade tides have been returned to an average of one marsh each year.

The various methods and approaches used are summarized in Table 11. The primary approaches have been tide-gate removal and replacement of undersized culverts. Some sites combined dredging with other activities; thus the total is more than 100%. Connecticut's experienced in-house staff and specialized low ground pressure equipment dedicated to tidal marsh restoration now allows the state to complete restoration projects at the lowest cost in all of New England (Louis Berger & Associates 1997).

Discussion

Vegetation

Our results demonstrate that returning tides to diked marshes initiates a pattern of decline by *Phragmites* or *Typha* and the reestablishment of tidal salt marsh vegetation. Based on rates of vegetation recovery, marshes fell into two groups that differed by an order of magnitude: ca. 0.5% (slower) and more than 5% (rapid) of total marsh area per year. Recovery was measured somewhat differently at IP1 and MC, but using the increase in *Spartina*-dominated vegetation these also fit with the rapid group.

Results from Barn Island IP1 (Sinicrope et al. 1990) and from the 1996 sites demonstrate that salinity is an important factor associated with this pattern of vegetation recovery. Salinity alone, however, cannot account for the dramatic difference in vegetation restoration rates. Seasonal pooled means from soil water wells were not significantly different among the five 1996 sites or between the two slower and three rapid sites taken together. The upper salinity limit for *Phragmites* survival is about 26‰. It can persist with moderate cover and shoot height at salinities slightly below this maximum; conversely, low salinities do not necessarily guarantee vigorous growth.

Hydroperiod, through its influence on soil redox potential and sulfide accumulation, appears to be another major factor influencing rates of *Phragmites* replacement by salt marsh angiosperms. Flooding frequency on both slower sites, GC and IP4, is constrained. At GC tide height is controlled by self-regulating tide gates, adjusted to minimize flooding of residential lots developed over 30+ years of reduced tidal prism. Most of the marsh area floods on just 15 to 20% of high tides, characteristic of higher high-marsh elevations (Bellet 2000), and *Phragmites* persists, although at reduced stem densities (ca. $25/m^2$) and heights (ca. 1 m). At IP4 tides are also damped relative to the open marsh immediately below the dike. Depth of tidal flooding at transect benchmarks for tides of known height (New London tide gauge corrected to Stonington [http://co-ops.nos.noaa.gov/data_res.html]) allowed hydroperiod estimates for mean transect elevations. Absolute elevations above the dike were the same or lower than below, but flooding frequency of the marsh surface below the dike and culvert was 2.0 to 2.9 times greater than above (28% vs. 10–15% of predicted growing season high tides reached or exceeded mean elevation of the transects). In addition, although most of IP4 remains dominated by *Phragmites*, a small area about 5 cm lower than the rest of the marsh (predicted flooding frequency ca. 20%) has converted to salt marsh vegetation.

The importance of hydroperiod is also supported by 1996 transect elevation data. In rapidly recovering systems the mean elevation of points with *Phragmites* cover more than 20% was significantly greater than points with low *Phragmites* cover. In contrast, on the two slowly recovering marshes there was no difference between the means of high and low *Phragmites* cover points. Also, on MC in the second growing season total angiosperm cover was sparse, with no elevation differences between points colonized by either *Phragmites* or *S. alterniflora.* However, on the same transect lines 5 years later *Spartina* had increased significantly, whereas its mean elevation fell; over the same period, *Phragmites* declined and became increasingly limited to higher less frequently flooded sites.

As might be predicted from reduced hydroperiods, mean depth to water table was greater at slower versus more rapidly recovering sites. Low redox potentials and high sulfide levels, soil chemistry parameters influenced by hydroperiod and soil water content, are the most probable environmental factors accounting for the sharp differential in the response of *Phragmites* and

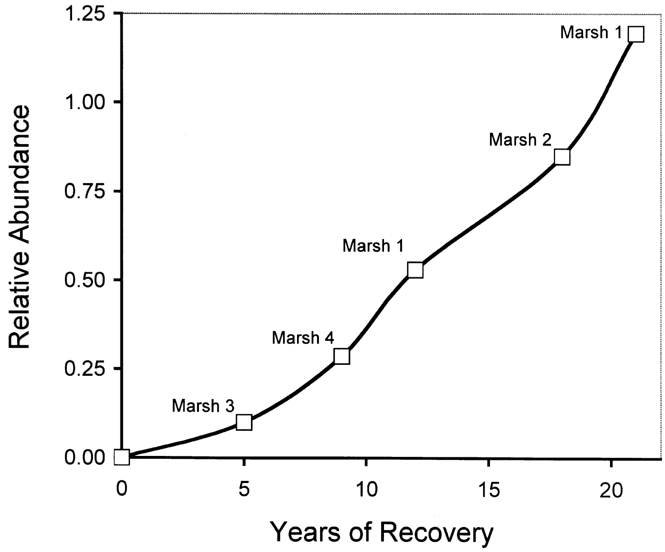

Figure 4. Relative abundance of *Melampus bidentatus* in recovering versus reference regions (mean density on restoration area/associated reference marsh) of four marshes at Barn Island in relation to the number of years of recovery. Although these marshes differ from one another in ways other than years of recovery, data indicate a long trajectory for full recovery of *Melampus* populations.

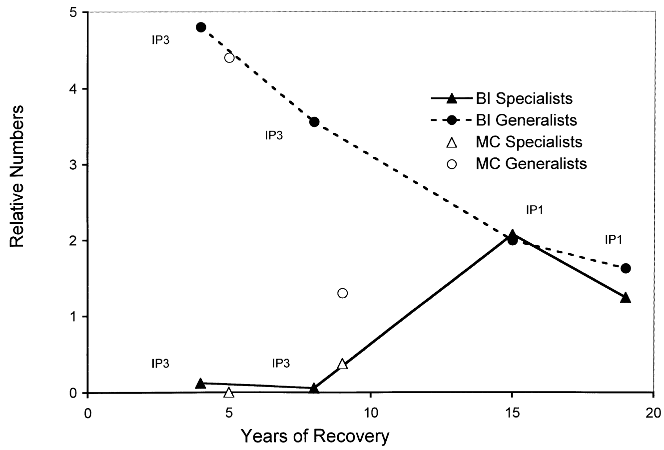

Figure 5. Relative abundance (recovering/reference) of birds considered salt marsh specialists (triangles) and salt marsh generalists (circles) at two recovering Barn Island (BI) marshes (solid) and at Mumford Cove (MC, open) plotted against years of restoration at the time counts were conducted. Although these marshes differ from one another in ways other than years of restoration, data indicate that it may take a decade for restoration sites to support equivalent populations of marsh specialists. Also, marsh generalists, whose use declines over time, rapidly occupy restoration sites.

the recovery of salt marsh vegetation among the 1996 sites (Hellings & Gallagher 1992; Chambers et al. 1998, 2002, in press; Bart & Hartman 2000).

Macroinvertebrates

It appears from the studies at Barn Island that certain members of the macroinvertebrate community may return in less than 5 years, relatively early in the restoration process. However, full recovery of some species, including *Melampus*, may be a slow process requiring two or more decades to achieve (Fig. 4). Collectively, results from all the recovering marshes considered in this report reveal two important points. First, populations of the various sampled species may recover at independent rates on a particular marsh, and recovery may occur more rapidly for certain species on some marshes than others. Second, macroinvertebrate population recoveries do not necessarily parallel vegetation change. For example, on Barn Island IP4, judged by the standard of *Phragmites* replacement to a slower recovering site, densities of *Philosica* and *Orchestia* were the same above and below the dike after 9 years but those of *Melampus* were not. In contrast, on IP1, where vegetation recovered rapidly, densities of *Philosica* and *Orchestia* were significantly lower than those on the reference site HQ, after 21 years, whereas *Melampus* populations appeared to have recovered fully. Comparison of invertebrate densities in the two slowly recovering 1996 marshes, as judged by vegetation, with the three more rapidly recovering sites further supports the contention that these designations do not carry over to the restoration of invertebrate populations and that various ecological attributes return at different rates.

Fishes

Less information is available on tidal marsh fishes than on the macroinvertebrates. It appears also in the case of fishes that characteristic species may return relatively early during restoration. A typical species assemblage occurred at MC 8 years after restoration. Similarly, at Barn Island essentially the same species of fish occurred in IP1 after 13 years of restored tidal flow as were present in the reference marsh below the dike and a nearby unimpounded valley marsh (Fell et al. 2000). However, *F. heteroclitus*, which was the numerically dominant species at all sites, appeared to be less abundant in IP1 than in the reference marshes. Eight years later, with different gear, as many *F. heteroclitus* were trapped above the dike as below (Swamy et al. 2002). Thus, although a typical species assemblage may return quickly after tidal restoration (Burdick et al. 1997; Roman et al. 2002, this issue), in some cases longer periods may be required for particular species to achieve numbers comparable with those of reference systems.

Thirteen years after the initial return of tides to IP1, gut content analysis of *F. heteroclitus* caught in mosquito ditches showed that the diets of this species were similar in IP1, the reference HQ, and unimpounded valley marshes. On the other hand, it appeared that *F. heteroclitus* caught in IP1 had consumed less food per unit body weight than had fish caught below the dike (Allen et al. 1994). Four years later differences in diet and gut fullness of *F. heteroclitus* trapped on the flooded marsh surfaces of IP1 and HQ were minimal, suggesting that the restored and reference marshes may be equivalent as foraging sites for this species. Further study, however, is required to settle this point. It is noteworthy that feeding on marsh surface invertebrates such as *Melampus*, *Orchestia*, and *Philosica* appears to be much less extensive at Barn Island than in some other Connecticut marshes sampled during the same time of year (Fell et al. 1998; Warren et al. 2001).

Birds

Benoit and Askins (1999) demonstrated that fewer bird species and a lower number of state-listed species use *Phragmites*-dominated marshes than comparable salt and brackish marshes that remain relatively *Phragmites* free. The results reported here demonstrate that recovering salt marsh vegetation on degraded tidal marshes will, over time, lead to recolonization of these sites by birds more uniquely associated with the habitat characteristics of *Spartina*-dominated tidal salt marshes.

Early stages of restoration (4 and 5 years at IP3 and MC, respectively) support a greater abundance and diversity of marsh generalists relative to the reference marsh, Barn Island HQ; habitat remained unsuitable for marsh specialists. After 9 and 10 years, however, Marsh Wrens were recorded at both sites and Saltmarsh Sharp-tailed Sparrow on MC; marsh generalists, however, were still twice as frequent as on HQ. Fifteen years after reintroduction of tides to IP1 (1995) *Spartina*-dominated salt marsh vegetation was well established; both Seaside and Saltmarsh Sharp-tailed Sparrows were nesting on the high marsh and were observed more frequently on IP1 than on the reference marsh (Brawley et al. 1998). The 1999 observations confirm a trajectory of increasing use of restored marshes by tidal marsh specialists such as Saltmarsh Sharp-tailed and Seaside Sparrows and at the expense of generalist species.

Willet, one of the least common marsh specialists and a Species of Special Concern in Connecticut, was not recorded at any of the restoration sites, although it does nest on HQ, immediately below IP1. Willet nests primarily in *S. patens* meadows; small patches of *S. patens* have developed on IP1, principally on the creek-bank levee, but it is still absent on IP3 and MC. Decades may be required before these marshes can support breeding populations of Willet.

Waders become more abundant in Connecticut marshes after June, and shorebirds migrate south in mid-summer, reaching their greatest numbers along the New England coast between July and September. Based on the 1994 and 1995 survey data it is clear that both long-legged waders and migrating shorebirds prefer the wetter more open restoration sites of IP1 and IP3 to the dryer HQ reference marsh (Brawley 1995). The substantially lower numbers of these species observed in 1999 most likely reflects the earlier, May and June, sampling compared with the 1994 and 1995 study (May to September).

Salt Marsh Restoration in Connecticut

The 1980 CMA was drafted specifically to emphasize restoration of degraded sites that once supported tidal wetlands, not marsh "creation." Although there was little peer-reviewed literature on salt marsh creation in 1980, the legislature's clear distinction between restoration and creation has proven appropriate. Success of salt marsh creation projects, often labeled "restoration" (Zedler & Callaway 1999) and proposed as mitigation in permit applications, has been problematic (Moy & Levin 1991; Zedler 1993; Simenstad & Thom 1996; Minello & Webb 1997; Zedler & Callaway 1999).

Office of Long Island Sound Program's management approach pragmatically recognizes that precise predictions on biotic community structure and ecosystem functions in tidally restored systems are unrealistic. Projects are not chosen or designed with expectations of recreating, precisely, conditions before tidal restriction, and in almost all cases managers allow natural processes to dictate ultimate form and function of tidally restored marshlands.

The DEP has supported research on a selected series of sites, the focus of this report, using the findings in an iterative process to assess restoration success and then design and implement new projects. Monitoring also contributes to adaptive management of restoration sites, helping to balance the goals of restoration with the political and social realities of intertidal back yards and flooded basements.

Summary and Conclusions

Many published reports that track the progress of tidal marsh restoration actually address created marshes, commonly established with the planting of *Spartina alterniflora*, *S. foliosa*, or some similar marsh dominant vegetation on bare substrate, usually dredged material or excavated upland. These studies have focused on vegetation establishment and development (Webb & Newling 1985; Broome et al. 1988; LaSalle et al. 1991;

Zedler 1993) and other ecosystem attributes, including soil nutrients (Craft et al. 1988), establishment of invertebrate and resident marsh fish communities (Moy & Levin 1991; Simenstad & Thom 1996; Minello & Webb 1997), and use by birds (Simenstad & Thom 1996; Zedler and Callaway 1999).

A critical question raised by some of these authors, either implicitly (Moy & Levin 1991; Minello & Webb 1997) or explicitly (Simenstad & Thom 1996; Zedler & Callaway 1999), is the validity of restoration "trajectories" (Aronson & LeFloc'h 1996; Hobbs & Norton 1996): the idea that physical conditions and ecological functions of restoration sites will follow temporal paths that approach and may eventually reach equivalence with comparable undegraded "reference" systems (Mitsch & Wilson 1996; Mitsch et al. 1998). Recently, Zedler and Callaway (1999) argued that if recovery trajectories exist at all, many ecologically important attributes and functions will not reach equivalence for several decades. In support of this position they cite their southern California marsh at Tijuana Bay, the Puget Sound site of Simenstad and Thom, created marshes in Galveston Bay studied by Minello and Web, and the North Carolina mitigation site reported on by Moy and Levin.

These are all marsh creation projects, however; substrates were less than ideal and in some hydroperiods were limited by excessive elevations (Minello & Webb 1997) or tidal restriction (Moy & Levin 1991). It should not be surprising that such sites would be very slow to reach parity with nearby reference marshes, which may be hundreds or thousands of years old. These authors are justified in their concern that such marsh creation sites may be used in permitting processes to mitigate marsh destruction.

Similar to the findings of several reports in this issue, such as Morgan and Short 2002; Thom et al. 2002; Tanner et al. 2002; and others, key tidal salt marsh functions and attributes for sites in our study do appear to be following restoration trajectories, bringing these sites within a range typical for Long Island Sound salt marshes within one to two decades. This reflects appropriate restored tidal hydrology (Burdick et al. 1997) and probably substrates—marsh peats, however modified by decades of tidal restriction, *Phragmites* or cattail growth, or burial by dredge material. It is important to stress that different attributes and functions (examples documented here and by Fell et al. [2000] include vegetation, high marsh macroinvertebrate populations, fish and bird use) recover at different and often independent rates. Even where hydroperiod limits rapid vegetation recovery, *Phragmites* height and density are continuing to decrease and salt marsh angiosperms expand, while macroinvertebrates and estuarine fish reestablish much more quickly. In large measure our findings support those of Mitsch et al. (1998) on the development of created riparian wetlands: Given appropriate substrate, hydrology, and available propagules, wetlands with community structures and ecological functions similar to natural systems will develop over time.

Two decades of results presented here address a range of ecosystem functions, attributes, and societal values associated with tidal salt marshes. Coupled with nonquantitative observations on a large number of additional restoration sites, these findings are consistent with the findings of Burdick et al. (1997) for northern New England, as well as most reports in this issue. They further strongly support our basic scientific hypothesis and Connecticut's management philosophy: Tides are the primary abiotic factor organizing tidal marsh communities. Returning tidal action will set degraded marshes on trajectories that will restore ecological attributes and functions and reconnect these wetlands to the larger estuarine-coastal ecosystem. Furthermore, it is unrealistic to target overly specific "final equilibrium" conditions. Marshes change over time without human input and trajectories suggest end points that may take many years to reach and a target reference marsh may change over such time frames as well. Full equivalence may take decades, but human biases about restoration time scales and eventual equilibrium communities are less important than reestablishing tidal connections between marshes, estuarine waters, and the larger coastal ecosystem.

Acknowledgments

The Connecticut DEP through grants from the Long Island Sound Research Fund and the Long Island Sound License Plate Fund supported this research. Additional support came from Connecticut College through Keck Undergraduate Research Fellowships, Mellow Student Summer Research Fellowships, the R. F. Johnson Faculty Research Fund, the Katherine Blunt and Jean C. Temple '65 endowed chairs, and summer housing provided for all undergraduate research assistants. The work and dedication of numerous undergraduate assistants contributed to this research, with particular thanks to Jessica J. Doe, Sheri A. Spiegal, Elisa M. Tarlow, Misha Body, Michael B. Keaney, Mawuli K. Nyaku, Rachel L. Avery, Elizabeth C. McIlvain, and Alice L. Keen. Jon Roy generously shared his 1997 Mumford Cove data from his graduate independent study course at Connecticut College.

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